



Can we save large carnivores without losing large carnivore science?

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1 Can we save large carnivores without losing large 2 carnivore science?

3

4 **Running title:** Saving large carnivore science

5

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41

42 **Abstract**

43 Large carnivores are depicted to shape entire ecosystems through top-down processes.
44 Studies describing these processes are often used to support interventionist wildlife
45 management practices, including carnivore reintroduction or lethal control programs.
46 Unfortunately, there is an increasing tendency to ignore, disregard or devalue fundamental
47 principles of the scientific method when communicating the reliability of current evidence for
48 the ecological roles that large carnivores may play, eroding public confidence in large
49 carnivore science and scientists. Here, we discuss six interrelated issues that currently
50 undermine the reliability of the available literature on the ecological roles of large carnivores:
51 (1) the overall paucity of available data, (2) reliability of carnivore population sampling
52 techniques, (3) general disregard for alternative hypotheses to top-down forcing, (4) lack of
53 applied science studies, (5) frequent use of logical fallacies, and (6) generalisation of results

54 from relatively pristine systems to those substantially altered by humans. We first describe
55 how widespread these issues are, and given this, show, for example, that evidence for the
56 roles of wolves (*Canis lupus*) and dingoes (*Canis lupus dingo*) in initiating trophic cascades
57 is not as strong as is often claimed. Managers and policy makers should exercise caution
58 when relying on this literature to inform wildlife management decisions. We emphasise the
59 value of manipulative experiments, and discuss the role of scientific knowledge in the
60 decision-making process. We hope that the issues we raise here prompt deeper consideration
61 of actual evidence, leading towards an improvement in both the rigour and communication of
62 large carnivore science.

63 **Keywords:** apex predator; behaviourally-mediated trophic cascades; adaptive management;
64 experimental design; mesopredator release hypothesis; science denial

65

66 **Introduction**

67 Large carnivores are some of the most charismatic and ecologically-influential organisms on
68 Earth. Through their interactions with other animals, large carnivores may affect faunal and
69 floral communities across multiple trophic levels ([Darwin 1859](#); [Leopold 1949](#); [Hairston et al. 1960](#)). This process is known as a trophic cascade ([Paine 1980](#)), and is a concept now
70 fully entrenched amongst ecologists, conservation biologists and many land and wildlife
71 managers.

73 Seldom have such novel ecological concepts been so rapidly mainstreamed to the extent that
74 they are identified as one of the 20 most influential topics in biodiversity conservation
75 ([Bradshaw et al. 2011](#)). Yet the ‘mesopredator release hypothesis’ (MRH) and its cousins the
76 ‘large-carnivore control-induced trophic cascade hypothesis’ (TCH) and the ‘behaviourally-
77 mediated trophic cascade hypothesis’ (BMTCH) have done exactly that, so much so that
78 these concepts are now routinely advanced as scientific and moral justification for what are
79 essentially highly normative standpoints concerning desired conservation outcomes.

80 Inherently value-laden, religious terms are now frequently used in academic discourses about
81 the ecological roles of large carnivores – terms such as hero, doctrine, dogma, demonising,
82 virtuous, saviour, scapegoat, sanctification, sinners and saints (e.g. [Jones 2002](#); [Soulé et al. 2005](#); [Anahita and Mix 2006](#); [Allen et al. 2011a](#); [Letnic et al. 2011](#); [Mech 2012](#); [Chapron and Lopez-Bao 2014](#); [Middleton 2014](#); [Johnson and Wallach 2016](#)). Unfortunately, but perhaps

85 motivated by the dire status of many carnivore populations, a growing number of studies rely
86 on weak inference to assess the roles of large carnivores in ecosystems (e.g. [Allen et al.](#)
87 [2013b](#); [Ford and Goheen 2015](#)). Such practices might stimulate short-term gains in carnivore
88 conservation and motivate some segments of the public to care about it, but these
89 communication practices risk undermining long-term confidence in large carnivore science
90 and scientists ([Fleming et al. 2012](#); [Sarewitz 2012](#); [Middleton 2014](#)). The actual *science* of
91 large carnivore science is getting lost, being replaced by catch phrases, slogans, sound bites,
92 YouTube clips, fake news and post-truth politics, or the simplification and popularisation of
93 unsubstantiated or unreliable theories and hypotheses. This tension between scientific rigour
94 and pursuit of quick conservation gain raises the critical question: can ecologists save large
95 carnivores without losing large carnivore science?

96 As described in several studies (summarised, for example, in [Crooks and Soulé 1999](#);
97 [Hayward and Somers 2009](#); [Terborgh and Estes 2010](#); [Eisenberg 2011](#); [Estes et al. 2011](#);
98 [Ritchie et al. 2012](#); [Ripple et al. 2014b](#); but for a clear definition see [Ripple et al. 2016b](#)), the
99 core theoretical processes associated with the MRH, TCH and BMTCH are:

- 100 1. Mesopredators and herbivores induce declines in smaller fauna and flora,
- 101 2. Large carnivores induce declines in mesopredators and herbivores,
- 102 3. Lethal control, harvest or hunting of large carnivores by humans induces declines in
103 large carnivores, increases in mesopredators and herbivores, and ultimately causes
104 undesirable outcomes for biodiversity and ecosystems,
- 105 4. Cessation of large carnivore control, harvest or hunting and/or active large carnivore
106 encouragement, including reintroduction, induces declines in mesopredators and
107 herbivores, which ultimately causes desirable outcomes for biodiversity and
108 ecosystems, and
- 109 5. Documentation of the MRH, TCH and BMTCH in some studies has been common
110 enough that these processes should be considered universal across ecosystem types
111 and independent of carnivore size or phylogeny.

112 The way these theories have been mainstreamed are perhaps best encapsulated in the short
113 online video titled *How wolves change rivers* ([Sustainable Human 2014](#)), which has been

114 viewed over 18 million times since early 2014, but which does not consider the contrary (and
115 often superior) evidence for the processes it claims. Proponents of the MRH, TCH, and
116 BMTCH argue that these hypotheses should be accepted by scientists and society as
117 ecological laws by default (not as mere theories or hypotheses) and that the burden of proof
118 for demonstrating their reality should be placed on those who do not believe them ([Estes et](#)
119 [al. 2011](#)). These theories also provide the scientific justification for many admirable and
120 worthwhile efforts to restore large carnivore populations to densities and distributions
121 reminiscent of former times ([Ripple et al. 2014b](#); [Ripple et al. 2016a](#)), although historical
122 ecological benchmarks have not been determined for most systems (e.g. [Hayward 2012](#)).
123 Nevertheless, the worldwide influence of the MRH, TCH and BMTCH have been enormous
124 ([Bradshaw et al. 2011](#)). In spite of the perceived universality of top-down control of
125 ecosystems however, there is a large and growing number of large carnivore studies
126 indicating that such effects are highly context specific and that many of the most rigorous
127 studies failed to document evidence of trophic cascades (Tables 1–3).

128 INSERT TABLE 1

129 INSERT TABLE 2

130 INSERT TABLE 3

131 In this brief overview, we summarise six key issues weakening the strength of the available
132 literature and undermining scientific advancement on understanding large carnivores’
133 ecological roles. We focus our discussion on grey wolves (*Canis lupus*) and Australian
134 dingoes (*Canis lupus dingo*), which have been claimed to be the only two terrestrial
135 carnivores for which both the MRH and TCH have been demonstrated (Figure S2 in [Ripple et](#)
136 [al. 2014b](#)). Our aim is not to denigrate these or other large carnivores, decrease interest in
137 them, diminish the motivation to conserve them, or hinder the pursuit of scientific knowledge
138 in this field. On the contrary, our aim is to outline the primary issues weakening the reliability
139 of research on MRH, TCH and BMTCH, to show why wildlife managers and policy makers
140 should exercise caution when making decisions based on the currently available literature
141 describing these processes. We agree with many authors that top-down forcing can occur and
142 that large carnivores can have important ecological roles. However, there are enormous gaps
143 in our understanding of when and where such effects will occur in most systems. Articulating
144 the truth about the reliability (or lack thereof) of large carnivore science is, in and of itself, a

145 strong conservation message: it is far better to err on the side of caution and preserve large
146 carnivores in the first place than to falsely believe ecosystems can be quickly and easily
147 fixed, restored or rewilded by simply bringing some carnivores back ([Glen et al. 2007](#);
148 [Marshall et al. 2016](#)). We further offer suggestions for overcoming these issues with the hope
149 that future large carnivore studies will avoid them and better contribute to the evidence-base
150 needed for the management and conservation of large carnivores and sympatric species.

151 **Issues that weaken the available literature supporting the MRH, TCH and BMTCH**

152 *1. There is not enough evidence of any kind, reliable or otherwise*

153 A general understanding of large carnivores' roles is only beginning to emerge, and much
154 more work is needed before we can confidently claim what those roles are or the ecological
155 contexts that shape these roles. Large carnivores unquestionably have ecological effects or
156 impacts of some description. In principle, every individual animal eaten by a carnivore
157 represents an impact – the prey animal is dead or scared, the prey's population growth or
158 foraging is slowed, scavengers scavenge, decomposers decompose, nutrients enter the soil,
159 life for the prey's competitor is now a little easier, the vegetation that would have been
160 consumed by the prey survives a little longer, and the carnivore lives to kill another day.
161 Whether the death of that prey animal is a good or bad thing (or not) depends on the
162 perspective of which animal is favoured over another ([Allen et al. 2011b](#); [Mech 2012](#)) – there
163 are winners and losers to every interaction ([Flagel et al. 2016](#)). These interactions all have a
164 value, contributing to the building blocks of wider ecological and demographic processes,
165 and evolutionary selection pressures ([Darwin 1859](#); [Hairston et al. 1960](#); [Kershaw 1969](#);
166 [Barbosa and Castellanos 2005](#); [Krebs 2008](#); [Molles 2012](#)). But do these individual-level
167 impacts of a relatively small magnitude combine and accumulate to produce detectable
168 cascading impacts of a large magnitude on populations and whole ecosystems? Are these
169 carnivore effects stronger or more important at shaping systems than bottom-up processes?
170 Can individual carnivores regulate entire food webs? Do carnivore effects always produce net
171 benefits to biodiversity? Are positive carnivore effects universal across ecosystems and
172 apparent across all trophic levels?

173 In spite of claims for the universality of trophic cascades and a concomitant shift in the
174 burden of proof to disprove top-down forcing and prove bottom-up forcing ([Terborgh and](#)
175 [Estes 2010](#); [Estes et al. 2011](#)), [Haswell et al. \(2017\)](#) shows that at best, detectably large

176 cascading effects of top-predators (from a wide range of taxonomic groups) are the exception
177 and not the rule. Indeed, several studies using strongly-inferential methods demonstrate that
178 such top-down effects do not always occur, or if they do, they are far weaker than bottom-up
179 processes (e.g. [Gasaway et al. 1983](#); [Boertje et al. 1996](#); [Hayes et al. 2003](#); [Vucetich and](#)
180 [Peterson 2004](#); [Vucetich et al. 2005](#); [Brodie and Giordano 2013](#); [Marshall et al. 2013](#); [Allen](#)
181 [et al. 2014b](#); [Ford et al. 2015a](#); [Sivy 2015](#); see also [Schmitz et al. 2000](#); [Bowyer et al. 2005](#);
182 [Sergio et al. 2008](#); [McCoy et al. 2012](#); [White 2013](#); [McPeck 2014](#); [Kuijper et al. In press](#)).
183 Ford and Goheen ([2015](#)) showed that of five strongly-inferential experiments investigating
184 large carnivores' roles, only two found evidence supporting the TCH. Morgan et al. ([2017](#))
185 highlight the supremacy of bottom-up processes and articulate the folly of attempting to shoe-
186 horn or apply outcomes from one ecological context into another. Recent global reviews of
187 the topic have also reported that 'little is known' about 24 of the 31 species of the world's
188 largest carnivores, as ecologists are only just beginning to discover their ecological functions
189 ([Ripple et al. 2014b](#)); or put another way, the MRH, TCH and BMTCH have not yet been
190 shown for at least 77% of large carnivores. Hence, we do not yet know what the ecological
191 functions of large carnivores are, and what we do know is from a minority of species in an
192 even smaller minority of biomes. While these hypotheses might eventually be applied to, and
193 supported in, a wide number of food webs, evidence supporting these hypotheses are
194 currently quite restricted.

195 Ripple et al. ([2014b](#)) claim that both the MRH and the TCH have been demonstrated only for
196 two related species, grey wolves and Australian dingoes, but the evidence-base for these two
197 species is very limited. In the case of dingoes, the total number of field studies on their
198 ecological roles is just a few dozen. Of these studies, all but four are observational or
199 correlative studies conducted in small areas (i.e. a few hundred km²) and over only a few
200 days ([Allen et al. 2013b](#); [Allen et al. 2015](#)). Drawing on this limited pool of empirical data,
201 the 22 literature reviews of dingoes' ecological roles produced over the last 10 years have
202 unavoidably borrowed heavily from each other in what might be called citation inbreeding
203 ([Allen et al. 2014c](#)). Thus, there is not a growing body of reliable evidence for dingoes'
204 ecological roles at all, but merely a growing body of largely recycled literature (Table 1; see
205 also [Allen et al. 2011b](#)). Evidence for the ecological roles of wolves is much stronger than
206 dingoes, but is still frequently challenged and often found unreliable for similar reasons
207 (Tables 2 and 3; see also [Winnie and Creel 2017](#)). The combination of mixed-outcomes when
208 testing the MRH, TCH and BMTCH and the absence of studies on most species of large

209 carnivore warrants far greater circumspection than is often afforded in syntheses of carnivore
210 ecology and conservation.

211 2. *Sampling methods for carnivores are often unreliable*

212 Studies measuring the effects of large carnivores roles typically correlate some change or
213 difference within an ecosystem to some change or difference in carnivore abundance ([Ford](#)
214 [and Goheen 2015](#)). But such approaches are frequently challenged because of their lack of
215 rigour (Tables 1–3). These challenges usually fall into three main categories of complaint:
216 experimental design constraints (e.g. manipulative experiments vs correlations or
217 observations; alternative hypotheses), predator sampling strategies (e.g. tracking plots,
218 camera traps, direct observations, movement data etc.), and data analysis approaches (e.g.
219 indices, occupancy modelling, statistical assumption violations, exclusion/inclusion of
220 outliers or contradictory data etc.). Counting or indexing carnivore populations can be
221 difficult and is often associated with large confidence intervals, but analytical methods do
222 exist to detect broad differences (e.g. [Kershaw 1969](#); [Caughley 1980](#); [Underwood 1997](#); [Zar](#)
223 [1999](#); [Quinn and Keough 2002](#); [Krebs 2008](#); [Engeman et al. 2017](#)). Unfortunately, many
224 studies use carnivore sampling methods that are incapable of yielding reliable data on
225 carnivore abundance, let alone actual rates of predation or perception of risk by prey animals.
226 The absence of these data undermines evidence for the proposed link between variation in
227 carnivore abundance and other reported changes and/or differences in the ecosystem.

228 Studies concluding that dingoes trigger trophic cascades are derived from non-validated and
229 often confounded comparisons of population indices between habitats, season, and/or species
230 ([Allen et al. 2011a](#); [Allen 2012b](#)). Ways to validate some common sampling methods have
231 been developed ([Allen and Engeman 2014](#)). When their methods are scrutinised, the results
232 of the most oft-cited works are unreliable ([Allen et al. 2014c](#)). Even the results of the best
233 available manipulative experiments are sometimes contested on grounds that the predator
234 sampling methods are unreliable (e.g. Table 1).

235 There is unlikely to ever be any one perfect predator sampling method that suits all
236 applications, so the use of different sampling techniques and analytical methods across
237 studies is not particularly concerning. It does not matter if carnivores are sampled using sand
238 plots, camera traps, snow tracking, GPS collaring, direct observations, or remote sensing (for
239 example) provided the data are subsequently handled and analysed appropriately. We argue

240 instead, that it is important to ensure that that whatever the implicit assumptions of the
241 methods are, that they are justifiable for the context under which the study was conducted
242 ([Engeman 2005](#); [Allen and Engeman 2014](#)). The weaknesses and limitations of these survey
243 methods need to be openly acknowledged and discussed – not only in the peer-reviewed
244 manuscript, but also in the subsequent public discourse. This is where many previous studies
245 have erred (Table 1), and where improvements must be made if science is to acquire less
246 ambiguous evidence to support the MRH, TCH or BMTCH ([Hayward et al. 2015](#)).

247 3. *Alternative hypotheses are seldom tested*

248 Carnivores are just one of many potential causal agents operating in ecosystems ([Vucetich et](#)
249 [al. 2005](#); [Middleton 2014](#); [Peterson et al. 2014](#); [Ford and Goheen 2015](#)). Yet for many studies
250 claiming support for the MRH, TCH and BMTCH, the study framework is designed to *create*
251 evidence for these hypotheses rather than being designed so that evidence for plausible
252 alternative hypotheses is both tested and compared at the same time ([Winnie 2014](#)). Studies
253 investigating these hypotheses commonly focus on competition, predation/removal or risk of
254 predation (Tables 1–3). But there are many more interaction types besides these within food
255 webs, which interaction types can also be strong and often do not conform to simple
256 expectations ([Muhly et al. 2013](#); [Saggiomo et al. 2017](#)). Invertebrate ([Meadows et al. 2017](#))
257 and theoretical (e.g. [Finke and Denno 2004](#); [Holt and Huxel 2007](#); [McCoy et al. 2012](#);
258 [McPeck 2014](#); [Kendall 2015](#)) studies highlight many different outcomes of predator removal
259 or addition, most of which have received little attention in the wider large carnivore literature
260 ([Fleming et al. 2012](#); [Mech 2012](#); [Ford and Goheen 2015](#); [Haswell et al. 2017](#)). The
261 consequence of not investigating plausible alternative explanations is that management
262 actions may completely overlook key processes contributing to declines of fauna (e.g. [Allen](#)
263 [2011](#); [Middleton et al. 2013b](#); [Cooke and Sorriquer 2017](#)), and they cannot discover these
264 processes because the study framework simply corroborates a narrow set of *a priori*
265 hypotheses without looking for others.

266 A clear example of the systemic failure to evaluate alternative hypotheses and ignore contrary
267 data comes from a series of studies conducted in the Greater Yellowstone Ecosystem, USA
268 ([Winnie 2014](#)). Environmental changes following the restoration of wolves to Yellowstone
269 National Park are often given as a clear example of the beneficial effects of restoring large
270 carnivores to ecosystems (Table 2), but there are alternative hypotheses to explain many of
271 the observed changes ([Vucetich et al. 2005](#); [Marshall et al. 2013](#); [Middleton et al. 2013b](#)).

272 There is strong evidence that wolves alone are not responsible for all the changes attributed to
273 them ([Mech 2012](#); [Winnie and Creel 2017](#)). Many other important changes to the
274 Yellowstone system occurred around the same time as wolf restoration, and ‘when we tell the
275 wolf story, we get the Yellowstone story wrong’ ([Middleton 2014](#)). Using data from 1961 to
276 2004, Vucetich et al. ([2005](#)) investigated the TCH and showed that changes in climate and
277 harvest rate are justified explanations for most of the observed decline in Yellowstone elk,
278 rather than heightened predation by wolves. Indeed, wolf predation was determined to be
279 compensatory to existing rates of mortality (e.g. from starvation or mortality from other
280 predators). In addition, early studies on the BMTCH reported that wolves scared herbivores
281 away from riparian areas, which reduced herbivory on trees and ultimately caused increased
282 tree growth ([Ripple and Beschta 2004](#); [Beschta and Ripple 2007](#)). Not only did these earlier
283 studies incorrectly identify areas of high predation risk ([Creel et al. 2005](#); [Kauffman et al.](#)
284 [2007](#); [Kauffman et al. 2010](#); [Winnie 2012](#)), but they also failed to consider more
285 parsimonious explanations for increased tree growth in riparian areas, such as the height of
286 the local water table ([Bilyeu et al. 2008](#); [Kauffman et al. 2013](#)). MacNulty et al. ([2016](#); pg.
287 27) summarise the present situation when they state that ‘scientific consensus about the role
288 of wolves in driving [trophic cascades] has yet to emerge, despite 20 years of research by
289 numerous federal, state and academic investigators’, and that the ‘overarching reason for the
290 impasse’ is the experimental design constraints on the Yellowstone wolf reintroduction
291 program. In other words, the lack of rigour and strong inference in testing the MRH, TCH
292 and BMTCH has generated the controversy over the role of wolves in restoring this
293 Yellowstone landscape.

294 In Australia, snap-shot studies comparing fauna abundances in adjacent areas separated by
295 predator-proof fences are commonly used to highlight the greater amount of biodiversity
296 present on the side of the fence with a greater number of dingoes (e.g. [Letnic et al. 2009](#);
297 [Fillios et al. 2010](#); [Letnic and Koch 2010](#); [Brawata and Neeman 2011](#); [Gordon et al. 2017a](#)).
298 However, the relative abundances of dingoes is not the only important difference between the
299 two sides of these fences (e.g. [Newsome et al. 2001](#); [Fitzsimmons 2007](#); [Allen 2011](#)). A
300 range of important geological and biophysical differences are also present, not the least of
301 which are the markedly different herbivore types, densities, and land-use histories, which are
302 also well-known to structure fauna communities through grazing-induced habitat changes
303 independent of dingoes or other predators ([Tiver and Andrew 1997](#); [Williams and Price 2010](#);
304 [Parsons et al. 2012](#); [Howland et al. 2014](#); [Koerner and Collins 2014](#)). The cross-fence

305 differences are obvious, but their causes are not. In spite of the appearance of a grandiose
306 ‘natural experiment’, the cross-fence comparisons are often poorly replicated and
307 confounded. Nonetheless, studies adopting this design have formed the bulwark of claims
308 about dingoes’ ecological roles ([Letnic et al. 2012](#); [Allen et al. 2013b](#); [Glen and Woodman](#)
309 [2013](#)). Until more rigorous experimental designs are implemented, further studies predicated
310 on correlative, cross-fence differences does little to increase evidence for the ecological role
311 of dingoes.

312 The management consequences of failing to address alternative hypotheses are exemplified
313 by the relatively simple carnivore system in Australia. Johnson and colleagues ([2007](#)) argued
314 that human control of dingoes in the last 200 years caused the continental collapse of
315 marsupial communities across Australia, but the role of the continental invasion of European
316 rabbits (*Oryctolagus cuniculus*; [Cooke and Soriguer 2017](#)) and the historical grazing of
317 introduced sheep (*Ovis aries*) coupled with drought ([Allen 2011](#)) were not properly assessed
318 as potential causal factors for marsupial decline. Johnson and colleagues continue to assert
319 that if only dingo persecution stopped, dingoes would suppress introduced rabbits, red foxes
320 (*Vulpes vulpes*) and feral cats (*Felis catus*), and facilitate the recovery of reintroduced
321 marsupials and other small mammals across the continent (e.g. [Johnson 2006](#); [Wallach et al.](#)
322 [2009](#); [Ritchie et al. 2012](#); [Letnic et al. 2013](#)). But such reintroductions continue to fail largely
323 because predators – including dingoes – keep quickly decimating reintroduced mammals
324 ([Christensen and Burrows 1995](#); [Moseby et al. 2011](#); [Bannister 2014](#); [Armstrong et al. 2015](#);
325 [Bannister et al. 2016](#)). All the dingoes occupying Australia did not prevent the historical
326 establishment and expansion of rabbits, foxes or cats across the continent in the first place,
327 nor did the presence of dingoes prevent the collapse of marsupial communities following the
328 advent of these pests. Extant dingo populations, never managed by modern humans across
329 roughly one-third of the Australian continent ([Allen et al. 2015](#)), have not facilitated
330 extirpation of these pests or their impacts, nor facilitated the recovery of marsupials in these
331 areas. Indeed, dingoes reach their highest densities in places with abundant rabbits ([Bird](#)
332 [1994](#); [Allen 2012a](#)), suggesting that invasive species are supporting carnivores rather than
333 large carnivores suppressing invasive species. Dingoes may even provide net benefits to
334 invasive rabbits through mesopredator suppression, just as dingoes putatively benefit rabbit-
335 sized native mammals ([Cooke and Soriguer 2017](#); [Gordon et al. 2017a](#)). In concert with
336 habitat changes (be these caused by livestock, fire or rabbits), dingo predation has been
337 identified as a key driver of native mammal decline independent of foxes or cats (e.g. [Kerle et](#)

338 [al. 1992](#); [Corbett 2001](#); [Lundie-Jenkins and Lowry 2005](#); [Barnes et al. 2008](#); [Allen 2011](#);
339 [Allen and Fleming 2012](#); [Allen and Leung 2012](#)). Yet dingoes are typically considered part of
340 the solution to Australia's fauna extinction crisis, when they are also part of the problem.
341 Continuing to ignore this and other alternative hypotheses wastes precious time in our
342 collective efforts to conserve native fauna under real threat of extinction.

343 In complex carnivore communities (where a wide variety of individual large carnivores
344 utilise a range of hunting strategies, resulting in increased heterogeneity in predator-prey
345 interactions), even manipulative experiments still struggle to tease apart the relative influence
346 of top-down and bottom-up processes (e.g. [Gasaway et al. 1983](#); [Boertje et al. 1996](#); [Maron
347 and Pearson 2011](#); [Sinclair et al. 2013](#); [Ford et al. 2015b](#); [Riginos 2015](#)). In fact, Riginos
348 (2015) goes as far as to suggest that behaviourally-mediated trophic cascades are either weak
349 or non-existent in African savanna systems because of the large sizes of many of the
350 herbivores (elephants, *Loxodonta africana*, in particular) and the over-riding effect of
351 climate. Predator diversity is known to dampen trophic cascade effects in model systems
352 ([Finke and Denno 2004](#)), and top-down forcing is also known to attenuate down through
353 trophic levels more rapidly than previously thought ([Schmitz et al. 2000](#); [Brodie et al. 2014](#)).

354 One characteristic of overemphasising the current robustness of large carnivore science is
355 ignoring, suppressing or omitting reference to alternative hypotheses and contrary data
356 ([Claridge 2013](#); [Winnie 2014](#)). This is easy for authors to do given the vast pool of citations
357 to choose from (e.g. Tables 1–3) and the limited number of references a journal will typically
358 accept. When accused of selective referencing, the plea of 'not enough room' (e.g. see [Marris
359 2014](#) for examples) does not promote objectivity and transparency. Rather, it disregards the
360 legitimate scientific criticisms available and only widens the creeping cracks of bias
361 described by Sarewitz (2012), who argued that research is riddled with systematic errors (see
362 also [Ioannidis 2005, 2014](#)) and that the ensuing debate then erodes public confidence in
363 science itself (see also [Fleming et al. 2012](#); [Middleton 2014](#)). Although large-scale and
364 observational 'natural experiments' have great value when their results are 'consistent with'
365 or 'inconsistent with' a given hypothesis, plausible alternative explanations nonetheless
366 require thorough exploration and ranking before reported results from 'natural experiments'
367 become the basis for changes in practice or policy ([Barley and Meeuwig 2016](#)). Investigating
368 alternative hypotheses should be a greater priority in future research on large carnivore
369 ecology.

370

371 4. *There is a dearth of applied-science studies*

372 Some research questions are largely academic (e.g. do species A and B have overlapping
373 diets?), whereas applied studies have direct and immediate relevance to land and fauna
374 managers (e.g. do interventions X and Y produce the same outcome for species A and B?).
375 The importance of understanding the ecological roles of large carnivores has implications for
376 the conservation and management of threatened carnivores and other fauna, such as livestock,
377 game, or threatened wildlife prey species (e.g. [Boertje et al. 2010](#)). Managers need
378 information that considers both the pros and cons of various management interventions, and
379 this is best achieved through manipulative experiments or adaptive-management studies that
380 investigate applied-science issues ([Glen et al. 2007](#); [Hone 2007](#); [Hone et al. 2015](#)). Questions
381 about the conservation utility of large carnivores as tools to restore biodiversity across the
382 landscape are answered much faster when truly applied questions are investigated.

383 Evidence for the effects of carnivore removal is also not the same thing as evidence for the
384 effects of their recovery (e.g. anisotropic vs isotropic effects; *sensu* [Ford and Goheen 2015](#)).
385 Simply re-establishing or bolstering large carnivores may not fix the many environmental
386 problems that occurred as a result of (and/or in addition to) carnivore extirpation ([Marshall et](#)
387 [al. 2013](#); [Marshall et al. 2014](#); [Wikenros et al. 2015](#)). In some cases, food web structure and
388 ecological context may have changed irreversibly (for whatever reason), some niches may no
389 longer exist, and a carnivore's function in the new ecosystem might now be different from
390 their previous function. Changes in the physical environment caused by the removal of large
391 carnivores may make the system resistant to complete restoration after large carnivores are
392 restored. This 'change resistant' hypothesis was tested against the existing TCH in a
393 replicated, randomized, manipulative experiment conducted over a decade. The hypothesis
394 that wolf restoration had caused ecosystem reorganization was rejected ([Marshall et al.](#)
395 [2014](#)), yet subsequent literature ignored it and instead repeated the story (i.e. [Sustainable](#)
396 [Human 2014](#)) that the ecosystems of Yellowstone have been dramatically restored by wolves
397 following their reintroduction. Restoring carnivore populations "to areas greatly modified by
398 human disturbance may not restore systems to their former state" ([Glen et al. 2007](#); pg. 498)
399 and these new carnivore functions may not be viewed as desirable or produce net benefits to
400 novel and still-changing ecosystems ([Fleming et al. 2012](#); [Flagel et al. 2016](#)).

401 Large carnivore studies often report a negative relationship between larger carnivores and
402 smaller or mesocarnivores, and are then quick to recommend wholesale changes to the way
403 large carnivores are managed without first measuring any actual effect of carnivore
404 management (e.g. hunting, removal, restoration) on large or small carnivores, herbivores or
405 prey (for examples, see [Letnic et al. 2009](#); [Wallach et al. 2010](#); [Colman et al. 2014](#); [Gordon](#)
406 [et al. 2017a](#); [Gordon et al. 2017b](#)). Equally, perceived negative impacts of carnivores on
407 livestock have historically been addressed by wholesale lethal control without any
408 recognition of the positive impacts that carnivores may have on the herbivores that compete
409 with livestock or the consequences of lethal control on livestock losses (e.g. [Wicks and Allen](#)
410 [2012](#); [Allen 2014](#); [Allen 2015a](#); [Prowse et al. 2015](#); [Allen 2017](#)). Treves et al. (2016) and
411 others (e.g. [Reddiex and Forsyth 2006](#); [Doherty and Ritchie 2017](#)) rightly point out that many
412 studies promoting predator control are badly designed, and we agree, but the same failing
413 exists in many studies condemning predator control and promoting predator conservation.
414 Unreliable science and poor science communication practices are a feature of literature
415 expressing both positive and negative views towards carnivores ([Boertje et al. 2010](#)).

416 To make ecological data useful for improving carnivore management and conservation,
417 researchers must provide managers with data they can apply. For example, when claiming
418 that large carnivore control (i.e. trapping, hunting, or poisoning) must be banned in order to
419 generate cascading, positive effects on biodiversity (e.g. [Carwardine et al. 2012](#)), information
420 on the actual effects of carnivore hunting or poisoning on biodiversity are needed, not just
421 information on how one carnivore species might interact with another (for examples, see
422 [Fleming et al. 2012](#); [Allen et al. 2015](#)). Conversely, when claiming that large carnivore
423 control must be implemented to reduce livestock predation, information on actual carnivore
424 impacts and impact reduction is required to ethically justify carnivore control ([Braysher](#)
425 [1993](#); [Allen et al. 2014b](#); [Allen 2017](#)). The paucity of applied ecological data in the wider
426 large carnivore literature means that much of the presently available information on the
427 MRH, TCH and BMTCH is not as useful to managers as it could be. This paucity also means
428 that, in most cases, we do not yet have a solid understanding of the actual cascading effects, if
429 any, of carnivore reintroduction, population control or manipulation ([Ripple et al. 2014b](#);
430 [Newsome et al. 2015](#)). This issue contributes to a significant knowledge-mobilization and
431 implementation gap for large carnivore science.

432 5. *Logical fallacies underpin much of the literature*

433 Most research about the ecological roles of large carnivores is also grounded in two logical
434 fallacies, *post hoc ergo propter hoc* and *cum hoc ergo propter hoc*. *Post hoc ergo propter hoc*
435 is the notion that if X occurred before Y, then X caused Y. When X is undesirable, this
436 pattern is often extended in reverse as: avoiding X will prevent Y. *Cum hoc ergo propter hoc*
437 is the notion that if X changed similarly to Y, then X and Y are linked. The fallacies lie in
438 coming to a conclusion based on the order or pattern of events, rather than accounting for
439 other factors that might rule out a proposed connection.

440 Examples of *post hoc ergo propter hoc* in the large carnivore literature are rife and include,
441 for example, conclusions to the effect that ‘the ecological changes observed in Yellowstone
442 National Park occurred after wolves were reintroduced, so wolves must have caused these
443 ecological changes’ (epitomised in [Sustainable Human 2014](#); see Table 2). Or alternatively,
444 ‘the last population of highly endangered mammals went extinct after predator control, so
445 predator control must have caused the extinction through trophic cascade effects’ (discussed
446 in [Fleming et al. 2013](#)). There are also many examples of *cum hoc ergo propter hoc*,
447 including almost all the relevant literature on dingoes’ ecological roles (see [Allen et al.](#)
448 [2013b](#); see Table 1). That wolves may not have been the cause of all the observed ecological
449 changes in Yellowstone since the mid-1990s is argued by Kauffman et al. ([2010](#)), Mech
450 ([2012](#)) and others (e.g. [Creel and Christianson 2009](#); [Winnie 2012](#); [Marshall et al. 2013](#);
451 [Marshall et al. 2014](#); [Middleton 2014](#); [Peterson et al. 2014](#); see Table 2). The long term study
452 of wolf–moose (*Alces americanus*)–habitat–climate relationships on Isle Royale illustrate the
453 difficulties of attributing cause and effect even in very simple ecosystems ([Vucetich and](#)
454 [Peterson 2004](#)). This case study stands out because researchers have explored multiple factors
455 at the same time, have been excessively cautious in the language they use to attribute
456 causality, and have constantly updated their views concerning the functioning of the
457 ecosystem as new data becomes available. Shifting the research focus from ‘trophic cascades’
458 to ‘food webs’ in this way can help overcome the subtle yet troublesome overreliance on
459 logical fallacies in studies of carnivores’ ecological roles ([Eisenberg et al. 2013](#)).

460 6. *Most of the ‘best evidence’ comes from ecosystems that do not represent the majority*
461 *of the earth’s surface or species*

462 Although there are still some large tracts of relatively intact land in some places, the reality is
463 that the majority of the earth’s surface has been substantially altered by humans, and
464 continues to be altered, in a modern epoch now labelled as the Anthropocene ([Zalasiewicz et](#)

465 [al. 2008](#); [Kueffer and Kaiser-Bunbury 2013](#)). Modern, human-dominated ecosystems
466 typically comprise mixed land-uses including urbanisation, forestry, mining, hunting,
467 recreation, agriculture (crops and/or livestock production) or other areas fragmented by roads,
468 railways and fences, and containing exotic plant and animal species and artificial water
469 sources ([Linnell 2011](#); [Fleming et al. 2012](#); [Mech 2012](#)). Most tests of the MRH, TCH and
470 BMTCH have occurred in relatively intact ecosystems with relatively minor human
471 footprints, such as the National Parks of Canada and the United States ([Hebblewhite et al.](#)
472 [2005](#); [Ray et al. 2005](#); [Hayward and Somers 2009](#); [Eisenberg 2011](#); [Kuijper et al. In press](#)).
473 Where studied, however, the strength and utility of carnivore effects on food webs in human-
474 modified systems appear dissimilar to those in less modified ecosystems (e.g. [Elmhagen et al.](#)
475 [2010](#); [Muhly et al. 2013](#); [Meadows et al. 2017](#); [Morgan et al. 2017](#)).

476 For example, the recolonization of wolves in Sweden resulted in widespread behaviour
477 change by humans in their moose (*Alces alces*) hunting practices that precluded, or at least
478 reduced, the anticipated numerical effects of wolves on moose ([Wikenros et al. 2015](#)).
479 “Because most of the worlds’ habitat that will be available for future colonization by large
480 predators are likely to be strongly influenced by humans..., human response behaviour may
481 constitute an important factor that ultimately may govern the impact of large predators on
482 their prey and thus on potential trophic cascades” ([Wikenros et al. 2015](#); pg. 18). This point is
483 further underscored by the situation in South Africa, where the introduction or removal of
484 large carnivores has largely been driven by economic incentives ([Lindsey et al. 2007](#)), and
485 the long term ecological effects have been overlooked. In Kenya, the indirect effect of
486 carnivores on tree communities was mediated by ranching practices and the spatial
487 distribution of cattle corrals ([Ford et al. 2014](#)). Comparative analyses of mammalian food
488 webs in protected areas versus human-dominated areas of Canada concluded that ‘human
489 influence on vegetation may strengthen bottom-up predominance and weaken top-down
490 trophic cascades in ecosystems’ and that ‘human influences on ecosystems may usurp top-
491 down and bottom-up effects’ ([Muhly et al. 2013](#)).

492 Theories about the effects of large carnivores on food webs, as developed in relatively
493 pristine areas, may not be readily transferable or applicable to the human-modified
494 landscapes that make up the majority of the Earth’s surface ([Haswell et al. 2017](#); [Morgan et](#)
495 [al. 2017](#)). This is because the direct and indirect effects of humans on all trophic levels may
496 simply overshadow any carnivore effects ([Muhly et al. 2013](#); [Darimont et al. 2015](#); [Clinchy](#)

497 [et al. 2016](#); [Kuijper et al. In press](#)). Carnivores are but one potential causal factor in a
498 multicausal world ([Vucetich and Peterson 2004](#); [Peterson et al. 2014](#); [MacNulty et al. 2016](#);
499 [Engeman et al. 2017](#)), and restoring large carnivores into these human-modified systems
500 without removing the many other, more important causal factors influencing biodiversity loss
501 is unlikely to succeed in reversing the situation ([Allen and Fleming 2012](#); [Fleming et al.](#)
502 [2012](#)). This is not to say that carnivore restoration efforts are unnecessary or should be
503 avoided ([Chapron et al. 2014](#)), but that we should more carefully consider the anticipated
504 benefits of these actions against the biophysical and anthropogenic factors that mediate the
505 top-down effects of carnivores.

506 **Implications for large carnivore science and management**

507 The prevalence of these six aforementioned issues in the literature on large carnivores
508 (Tables 1–3) underscores our contention that evidence for the MRH, TCH and BMTCH is
509 undeniably weaker than is often claimed in journal articles or public discourse. Syntheses and
510 literature reviews of large carnivores' ecological roles should identify these issues, but they
511 usually do not, instead routinely failing to assess the internal validity of the original studies
512 reviewed, as described by Bilotta et al. ([2014](#)). When the individual empirical studies that
513 make-up the content of these reviews are judged against Platt's ([1964](#)) criteria for strong
514 inference, Hone's ([2007](#)) deconstruction of experimental design capabilities, or Sutherland
515 and colleagues' ([2013](#)) 20 tips for interpreting scientific claims, it is clear that even literature
516 reviews (e.g. [Ritchie and Johnson 2009](#); [Estes et al. 2011](#); [Ripple et al. 2014b](#)) seldom offer
517 reliable guidance on the state of the literature addressing the MRH, TCH and BMTCH. These
518 remain intriguing hypotheses, but they are each inadequately tested and not yet demonstrated
519 for almost all large carnivores and contexts.

520 We fear that the debates about the issues we raise here (Tables 1–3) are heading towards the
521 type of science denialism that plague medicine or climate science (see [Diethelm and McKee](#)
522 [2009](#)). In a growing number of cases, strong evidence against MRH, TCH and BMTCH is
523 denied while promoting these hypotheses using tactics common to science denial in other
524 disciplines, such as selectivity, use of logical fallacies, disregard of experimental work, and
525 deference to correlations (for examples, see [Letnic et al. 2011](#); [Ripple et al. 2011](#); [Beschta et](#)
526 [al. 2014](#); [Forsyth et al. 2014](#); [Johnson et al. 2014](#); for responses, see [Hodges 2012](#); [Squires et](#)
527 [al. 2012](#); [Fleming et al. 2013](#); [Allen et al. 2014a](#); [Winnie 2014](#); [Allen and West 2015](#)).
528 Science denialism is often characterised by downplaying the scope of a threat ([Russell and](#)

529 [Blackburn 2017](#)). In the field of large carnivore science, this is clearly manifest in claims that
530 carnivores are not a major problem for livestock producers or game ranchers (e.g. [Forsyth et](#)
531 [al. 2014](#)). It is also manifest in claims that native large carnivores will suppress unwanted
532 exotic species while denying that the same native carnivores can also suppress the threatened
533 native species they are assumed to provide protection for (see [Fleming et al. 2013](#) or [Allen](#)
534 [and Fleming 2012](#) for discussion). Dismissing or downplaying the legitimacy of scientific
535 criticisms as mere ‘controversy’ or ‘debate’ (e.g. [Ritchie et al. 2014](#); [Newsome et al. 2015](#)) is
536 also a form of passive science denialism. In truth, carnivores can have direct and indirect
537 positive, negative or neutral impacts on social, economic and environmental values, and these
538 impacts can change from time to time and place to place ([Chamberlain et al. 2014](#); [Haswell et](#)
539 [al. 2017](#)). But emphasizing ‘the good’ while downplaying ‘the bad’ only produces ‘the ugly’
540 literature on carnivore science, while also fostering the rise of invasive species science
541 denialism ([Russell and Blackburn 2017](#)). Such post-truth incredulities over evidence risks
542 reversing progress in a field that is tackling some of the most important and engaging
543 questions in modern ecology – namely, how does society restore and coexist with large fauna
544 in human-occupied landscapes ([LaRue et al. 2012](#); [Chapron et al. 2014](#)) and what may be the
545 ecological outcomes of this restoration effort?

546 Debates about the scientific understanding of, and appropriate management response to, large
547 carnivore impacts are not new. For example, in Alaska and northern Canada there has been
548 an ongoing debate about the impact of wolf and grizzly bear (*Ursus arctos*) predation on
549 moose and caribou (*Rangifer tarandus*) populations for decades (e.g. [Orians et al. 1997](#);
550 [Kennedy and Fiorino 2011](#)). The discourse has centred on the extent to which lethal control
551 of wolf and bear populations will lead to an increase in the harvestable surplus of moose and
552 caribou. An enormous amount of intensive research, of both descriptive and experimental
553 types (reviewed by [Boertje et al. 2010](#)), has been conducted in the region since the 1970’s
554 with the aim of understanding predator-prey relationships. But just like the Yellowstone
555 region ([MacNulty et al. 2016](#)), there is still huge uncertainty and controversy about the nature
556 of these trophic interactions and their consequences for management despite this considerable
557 research investment (e.g. [Van Ballenberghe 2006](#); [Boertje et al. 2010](#); [Kennedy and Fiorino](#)
558 [2011](#)). Lessons that can be extracted from this ongoing saga include: (1) even with massive
559 investment in research over many decades in relatively simple ecosystems it can still be a
560 challenge to understand the nature of interactions between predators and prey, let alone the
561 wider ecosystem impacts of human intervention on lower trophic levels; (2) valuable insights

562 can be obtained by exploring such relationships through the lens of predator-prey theory and
563 demographic models, an approach which has been almost absent from the recent generation
564 of trophic cascade studies (Tables 1–3); and (3) competing scientific results can rapidly be
565 included into what are essentially value debates about different worldviews. The maturation
566 of this controversy clearly shows how important it is to be aware of the intrinsic uncertainty
567 and context-dependence (in time and space) of any research results, and of the need to clearly
568 distinguish science from values in policy debates.

569 There are, of course, studies that are not encumbered by the six issues we raise, studies that
570 do indeed provide strong support for the MRH, TCH and BMTCH. Much of this can be
571 found in literature from marine, aquatic and invertebrate systems ([Heath et al. 2014](#);
572 [Meadows et al. 2017](#)), or systems and models where bottom-up processes are relatively
573 predictable, stable and controllable. Reliable work on MRH, TCH and BMTCH in terrestrial
574 systems is only beginning to catch up to these disciplines. Literature reviews and syntheses
575 are important as the field develops, but as described above, most of the reviews presently
576 available are inadequate. There is, therefore, an urgent need for a systematic review (*sensu*
577 [Pullin and Knight 2009](#)) of terrestrial studies that have used only manipulative experiments to
578 investigate these hypotheses— experiments inclusive of paired treated and non-treated areas,
579 sampled before and after treatments (e.g. carnivore removal or addition) over sufficient
580 temporal and spatial scales to detect cascading responses of predators, prey and plants. A
581 systematic review of such experimental studies, which excludes low-inference studies and
582 summarises the results of only those with the actual capacity to assess causal processes, may
583 produce useful insights into underlying ecological processes and be of great value to
584 carnivore managers ([Pullin and Knight 2009](#); e.g. [Boertje et al. 2010](#)). It would also yield
585 lessons on how to do more such research on different species, and in different contexts.

586 Many authors have called for such large-scale, long-term manipulative experiments
587 investigating the removal or addition of large carnivores (e.g. [Glen et al. 2007](#); [Ritchie et al.](#)
588 [2012](#); [Newsome et al. 2015](#)). Although such experiments are expensive and difficult to
589 achieve because of the logistical challenges arising from the massive scales that large
590 carnivores utilise, they can and have been done in some places (e.g. [Eldridge et al. 2002](#);
591 [Hayes et al. 2003](#); [Hebblewhite et al. 2005](#); [Allen et al. 2013a](#); [Marshall et al. 2013](#); [Allen et](#)
592 [al. 2014b](#); [Christianson and Creel 2014](#); [Ford et al. 2014](#); [Hervieux et al. 2014](#); [Ford et al.](#)
593 [2015b](#); [Mitchell et al. 2015](#)). These have often, but not always, shown support for elements of

594 the MRH, TCH and BMTCH; less so for dingoes ([Allen et al. 2014b](#)) but more so for wolves
595 ([Winnie and Creel 2017](#)). It is unlikely that many large carnivores will be subject to
596 experimental studies like these, or like the famous Kluane project on the Canada lynx (*Lynx*
597 *canadensis*) and snowshoe hare (*Lepus americanus*) system ([Krebs et al. 2001](#)). As a
598 consequence, it is highly unlikely that we will ever have access to knowledge from such
599 experiments for most large carnivores. Thus, a systematic review of studies testing the MRH,
600 TCH and BMTCH with only strongly-inferential methods will be all the more valuable. It
601 must also be remembered that while well-designed and implemented experiments will greatly
602 advance our understanding of theoretical ecological principles ([Engeman et al. 2017](#)), the
603 portability of their results may still be limited ([Schmitz et al. 2000](#); [Haswell et al. 2017](#);
604 [Morgan et al. 2017](#)).

605 Our focus on improving research rigour is not intended to imply that observational or
606 correlative ecological studies are not useful. Such studies are absolutely crucial to capture the
607 broad spatial and temporal dynamics over which large carnivores and their prey interact
608 ([Barley and Meeuwig 2016](#)). However, we argue that researchers need to exercise a greater
609 degree of caution in the interpretation and communication of studies on the MRH, TCH and
610 BMTCH, no matter how they are designed and conducted, and especially when they are used
611 as the basis for radical changes in carnivore management and policy – including cases where
612 lethal control and reintroduction are used. The associated biases, uncertainties, and ability to
613 make inferences need to become ever more central parts of the communication of research
614 results ([Johnson et al. 2015](#)). While we hope that scientists should manage this within the
615 pages of peer-reviewed journals, additional challenges arise when trying to communicate
616 uncertainty to the wider public ([Dixon and Clarke 2013](#)). In such contexts it is normally
617 impossible to successfully communicate such intrinsic limitations, making it all the more
618 important that authors take extreme care to not oversell the generality of their findings, nor
619 allow others to do so, and clearly separate between scientific findings and the various
620 normative policy or management action contexts within which these findings might be
621 operationalised.

622 The reality is that the knowledge available to wildlife managers will at best be limited to a
623 solid understanding of the natural history and ecology of the predators, their prey, and the
624 ecosystem, and based largely on data derived from time series, cross-site comparisons,
625 ‘natural experiments’ or other correlative studies ([Barley and Meeuwig 2016](#); [MacNulty et al.](#)

626 [2016](#)). A good understanding of species ecology can serve to exclude spurious or
627 unreasonable interpretations of correlative data, and such studies can also exclude certain
628 hypotheses or provide indirect support for other hypotheses for which experiments could be
629 designed to provide a definitive test (e.g. [Platt 1964](#); [Kershaw 1969](#); [Underwood 1997](#);
630 [Fairweather and Quinn 2006](#)). While these types of lower-inference studies may not
631 overcome all the aforementioned issues we describe, they do have the advantage of being far
632 cheaper and faster to conduct under a wide range of different ecological conditions, which
633 can address problems associated with the transferability of knowledge between contexts.
634 Ideally, conservation actions should be monitored within an adaptive management system
635 that can be used to permit the study of system responses to specific management
636 interventions ([Fleming et al. 2014](#); [Johnson et al. 2015](#)). This provides insights into how the
637 system functions and how management actions produce outcomes. Certain forms of carefully
638 designed adaptive management exercises can even be viewed as quasi-experiments ([Williams
639 and Brown 2014](#); [Johnson et al. 2015](#)).

640 Given the perilous conservation situation of many large carnivore species, there is a clear
641 need to act based on the best available knowledge at any given time ([Ripple et al. 2016a](#)).
642 However, manipulative experiments clearly trump anecdotal, observational and/or correlative
643 information for their informative value, and should therefore be valued more highly in the
644 decision making process ([Platt 1964](#); [Fleming et al. 2013](#)). While the weight of evidence for
645 the general role of large carnivores in triggering trophic cascades is indeterminate at this time
646 (but we look forward to this potentially changing one day), we caution researchers and
647 science communicators to carefully consider the implications of simultaneously advocating
648 for both large carnivore conservation and the primacy of top-down trophic cascades. These
649 two forms of advocacy need not be linked – carnivore conservation can often be justified on a
650 number of moral, ethical, and existential grounds that have nothing to do with trophic
651 cascades. At one extreme, such advocacy may contribute towards specious reintroduction
652 efforts that divert funds from broader conservation goals and/or place the livelihoods of local
653 people at risk ([Ford et al. 2017](#)). On the other extreme, we recognize that there will be no
654 perfect study to ever ‘close the book’ on the prevalence of trophic cascades, regardless of
655 their occurrence in nature. Because strongly-inferential, long-term, manipulative studies will
656 be difficult to implement in a cost-effective and timely manner to support these decisions, we
657 argue that knowledge of trophic cascades must be considered in management deliberations
658 but should not necessarily determine their outcome.

659 Whether or not society should or shouldn't restore large carnivores is outside the scope of our
660 present analysis (but see [Lewis et al. 2017](#)), and in the end, how large carnivores are
661 managed is a judgement that society must make, and which will largely be based on which
662 species (predator or prey or human interest) is given priority over another. In the Canadian
663 case of Hervieux et al. ([2014](#)), for example, the immediate interests of ungulates were
664 ultimately favoured over those of the wolves. Whereas, in the familiar Yellowstone story
665 (e.g. [Middleton 2014](#)), the interests of wolves were ultimately favoured over those of the
666 ungulates. Whether large carnivores are viewed as a 'good thing' or a 'bad thing' for an
667 ecosystem largely rests on the attention given to which species (livestock, invasive pests,
668 game species or threatened native fauna) carnivores happen to be killing at the time ([Allen et
669 al. 2011b](#); [Mech 2012](#)). As carnivore conservationists ourselves, we relish any excuse to
670 promote their conservation and recovery where it is needed and possible. But as scientists, we
671 lament the lack of objectivity and critical thinking underpinning the current 'parental
672 affection' (*sensu* [Chamberlin 1890](#)) towards the MRH, TCH, and BMTCH and the extent to
673 which this affection is used to legitimise selected views on carnivore management.

674 Upon reflection, we also observe that debates about large carnivore management (Tables 1–
675 3) are often not so much about differing beliefs or views about carnivores' actual functional
676 roles, but more so about the quality of scientific evidence people are willing to accept. Large
677 carnivore conservation is a bold and historically-novel judgement which must inevitably be
678 made on incomplete ecological evidence. Ecological evidence alone is insufficient to make
679 decisions, which must also account for the ethical, cultural and socio-political factors that
680 shape decision making in society (e.g. [Van Ballenberghe 2006](#); [Mech 2010](#); [Trouwborst
681 2010](#); [Fleming et al. 2014](#); [Olson et al. 2015](#); [Trouwborst 2015](#); [Marshall et al. 2016](#); [Lewis
682 et al. 2017](#)). We hope that the issues we raise here prompt deeper consideration of actual
683 evidence, leading to an improvement in both the rigour and communication of large carnivore
684 science, because the fates of many large carnivores and the integrity of associated ecological
685 processes are depending on it.

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1364 **Tables**

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1367 Table 1 – Some recent lines of debate discussing large carnivores’ roles in trophic cascades in
 1368 Australia, demonstrating that evidence for the ecological roles of dingoes is equivocal,
 1369 primarily due to the six issues described in the present article.

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Debated topic	Chronological order	Reference
Trophic cascades following dingo control	1	Wallach and O'Neill 2009
	2	Allen 2010
Ecological niche of dingoes	1	Fleming <i>et al.</i> 2012
	2	Johnson and Ritchie 2013
	3	Fleming <i>et al.</i> 2013
	4	Claridge 2013
Dingo predation risk to fauna	1	Dickman <i>et al.</i> 2009
	2	Allen and Fleming 2012
Methodological problems with dingo studies	1	Allen <i>et al.</i> 2011a
	2	Letnic <i>et al.</i> 2011
	3	Allen <i>et al.</i> 2011b
	4	Glen 2012
	5	Allen <i>et al.</i> 2013b
Cause of historical declines of marsupials	1	Johnson <i>et al.</i> 2007
	2	Allen 2011
Importance of dingo social structure	1	Wallach <i>et al.</i> 2009
	2	Allen 2012b
Trophic cascades following dingo control	1	Colman <i>et al.</i> 2014
	2	Allen 2015b
	3	Colman <i>et al.</i> 2015
Effects of dingoes on sheep	1	East and Foreman 2011
	2	Allen and West 2013
	3	Forsyth <i>et al.</i> 2014
	4	Allen and West 2015
Trophic cascades following dingo control	1	Allen <i>et al.</i> 2013a
	2	Johnson <i>et al.</i> 2014
	3	Allen <i>et al.</i> 2014a
	4	Allen <i>et al.</i> 2014b
	5	Hayward and Marlow 2014
	6	Nimmo <i>et al.</i> 2015
	7	Hayward <i>et al.</i> 2015

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1379 Table 2 – Some recent lines of debate discussing large carnivores’ roles in trophic cascades in
 1380 North America, demonstrating that evidence for the ecological roles of wolves is equivocal,
 1381 primarily due to the six issues described in the present article.
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Debated topic	Chronological order	Reference
Wolf-induced behaviourally-mediated trophic cascades in Yellowstone	1	Ripple and Beschta 2004 ¹³⁸³
	2	Kauffman <i>et al.</i> 2007
	3	Ripple and Beschta 2007 ¹³⁸⁴
	4	Kauffman <i>et al.</i> 2010
	5	Kimble <i>et al.</i> 2011
	6	Winnie 2012
	7	Beschta and Ripple 2013
	8	Kauffman <i>et al.</i> 2013
	9	Middleton <i>et al.</i> 2013a
	10	Beschta <i>et al.</i> 2014
	11	Winnie 2014
	12	Painter <i>et al.</i> 2015
Willow recovery in Yellowstone following wolf reintroduction	1	Ripple and Beschta 2003
	2	Despain 2005
	3	Ripple and Beschta 2006
	4	Wolf <i>et al.</i> 2007
	5	Beyer <i>et al.</i> 2007
	6	Bilyeu <i>et al.</i> 2008
	7	Creel and Christianson 2009
	8	Tercek <i>et al.</i> 2010
	9	Johnston <i>et al.</i> 2011
	10	Middleton <i>et al.</i> 2013a
	11	Marshall <i>et al.</i> 2013
	12	Marshall <i>et al.</i> 2014
	13	Smith <i>et al.</i> 2016
Trophic cascades and Mexican wolves	1	Beschta and Ripple 2010
	2	Mech 2012
Wolf effects on lynx	1	Ripple <i>et al.</i> 2011
	2	Hodges 2012
	3	Squires <i>et al.</i> 2012
	4	Wirsing <i>et al.</i> 2012
Wolf effects on bears	1	Ripple <i>et al.</i> 2014a
	2	Barber-Meyer 2015
	3	Ripple <i>et al.</i> 2015
Ethics and effects of predator control for moose conservation in Alaska	1	WMRC 1996
	2	Orians <i>et al.</i> 1997
	3	Van Ballenberghe 2006
	4	Boertje <i>et al.</i> 2010
	5	Kennedy and Fiorino 2011

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Table 3 – Some recent lines of debate discussing large carnivores’ roles in trophic cascades in Europe, demonstrating that evidence for the ecological roles of large carnivores is equivocal, primarily due to the six issues described in the present article.

Debated topic	Chronological order	Reference
Human influence on trophic cascades in Europe	1	Melis <i>et al.</i> 2009
	2	Kuijper 2011
	3	Dorresteijn <i>et al.</i> 2015
	4	Kuijper <i>et al.</i> 2016
	5	Ritchie <i>et al.</i> 2016
Large carnivore impacts on mesocarnivores	1	Palomares <i>et al.</i> 1995
	2	Palomares <i>et al.</i> 1998
	3	Sunde <i>et al.</i> 1999
	4	Linnell and Strand 2002
	5	Helldin <i>et al.</i> 2006
	6	Elmhagen and Rushton 2007
	7	Kowalczyk <i>et al.</i> 2009
	8	Pasanen-Mortensen <i>et al.</i> 2013
	9	Wikenros <i>et al.</i> 2014
	10	Pasanen-Mortensen and Elmhagen 2015

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